Responses of Songbirds and Small Mammals to Harvests of Native Grasslands for Biofuels in Western Minnesota

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Robert Michael Dunlap

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May the meadowlark's song never be forgotten.

Abstract

Some grassland birds and small mammals exhibit changes in abundance following vegetation removal in the previous year, but it is unknown to what extent these organisms respond to harvests of diverse, native grasslands. This thesis examines the effects of harvesting such grasslands on songbirds and small mammals, representing an important step in evaluating the conservation value of grasslands used for biofuel harvest. I analyzed abundance data collected from area-based bird surveys in grassland biofuel plots harvested via different pattern and percentage in western Minnesota, USA, from 2009 to 2013 (Chapter 1). Small mammal trapping was conducted in the same plots from 2009 to 2012, and abundance and occupancy data collected from these surveys was also analyzed (Chapter 2). I estimated relative abundance of 11 species of grassland birds and 7 species/genera of small mammals among the different harvest intensities and years of study. Four bird species and species richness showed declines in abundance following harvests, whereas two species showed increases in abundance. Harvests also resulted in negative impacts on two small mammals. The removal of vegetation in fall results in shorter, less dense vegetation the following spring, which creates largely unsuitable habitat for tall-grass songbirds (e.g., sedge wren) but more optimal habitat for species that prefer shorter vegetation (e.g., grasshopper sparrow). Additionally, the reduction in ground litter is detrimental to small mammals that prefer thicker vegetation (e.g., voles of the genus *Microtus*). At the community level, harvesting native grasslands appears to have little impact on grassland birds and small mammals, but it is nonetheless important to identify what species are present prior to harvesting so that harvesting activities do not result in detriment to these species.

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Preface

Biofuels offer a renewable source of combustible energy to nations searching for alternatives to finite sources such as oil. While biofuels such as ethanol have been available for decades, it is only recently that nations such as the United States have seriously considered grassland biofuels as a significant energy source of alternative energy. Many grass species have now become popular as biofuel crops in the United States, with switchgrass (*Panicum virgatum*) being among the most commonly planted (Keshwani and Cheng 2009, Hartman et al. 2011).

Ideally, however, the plant species sown and harvested would be representative of native plant communities rather than switchgrass monocultures such that these biofuel grasslands might provide quality habitat for native wildlife (Fletcher et al. 2010). Switchgrass monocultures are not the ideal state of native grassland habitat in the United States, nor did they comprise the original landscape of American prairies. Heterogeneity is required if restored grasslands are to provide the greatest benefit to obligate grassland wildlife (Hartman et al. 2011). Some species of grassland birds and small mammals have exhibited population changes in response to harvests of monoculture grasslands (Murry and Best 2003, Semere and Slater 2007), but it is unknown to what extent these same organisms respond to harvests of native, more diverse grasslands.

In 2009, researchers at the University of Minnesota's Cedar Creek Ecosystem Science Reserve began studying native grassland vegetation for its potential as both wildlife habitat and a competitive source of alternative energy (Williams et al. 2013). In addition to studying biofuel yields from the harvested vegetation (Jungers et al. 2013), researchers were interested in identifying the effects of harvesting on birds, small mammals, reptiles and amphibians, insects, and flowering plants. The experimental design included a total of 59 grassland plots averaging 8 ha (20 acres) in size in three regions of western Minnesota. All plots consisted of restored prairie under federal, state,

or private management. Plots within each region were chosen by referencing maps that included such parameters as soil type, elevation, wetland delineation, and land cover and then assessed for suitability via site visits (Williams et al. 2013).

Plots were randomly assigned one of six harvest treatments: control (no harvest), 50% block harvest, 50% strip harvest, 75% block harvest, 75% strip harvest, and 100% harvest. The varying plot treatments were designed to test whether the amount and pattern of vegetation harvested were important to grassland wildlife (Williams et al. 2013). Wildlife surveys were first conducted in 2009 to establish control conditions in the plots prior to the first harvest later that same year. Each year thereafter, all surveys were considered post-harvest surveys as they followed harvest the previous fall. For more information on survey methods associated with the various wildlife surveys conducted, see www.cbs.umn.edu/wildlife.

This thesis examines the bird and small mammal portions of the project and comprises two manuscripts formatted for publication in peer-reviewed journals. Chapter 1, "Grassland Songbird Responses to Biofuel Harvests in Western Minnesota," is intended for publication in American Midland Naturalist and is formatted as such. Chapter 2, "Short-term Effects of Native Grassland Biofuel Harvests on Small Mammals in Western Minnesota," is intended for publication in the Journal of Wildlife Management and is likewise formatted as such. Because both manuscripts will be multi-authored, I have used plural pronouns throughout; however, this thesis represents my own analysis and writing, and I am entirely responsible for its content.

CHAPTER 1: GRASSLAND SONGBIRD RESPONSES TO BIOFUEL HARVESTS IN WESTERN MINNESOTA

Grassland bird populations have declined throughout North America as native grasslands continue to be replaced and degraded by agricultural activities (Herkert, 1994; McCracken, 2005; Mineau and Whiteside, 2013). One potential opportunity to help remedy this is to increase the amount of land suitable as habitat for prairie birds by planting native grassland vegetation for biofuel harvest. Native grasses and forbs can be planted on agriculturally poor land, which means that farmers could potentially profit off land not currently suitable for crops (Fletcher *et al.*, 2011). Researchers have estimated that native grassland vegetation can compete with corn and soybeans in overall biofuel yield (Tilman *et al.*, 2006; Jungers *et al*., 2013). Additionally, diverse vegetation communities not only promote greater biodiversity but are also more resistant to species-specific diseases and exotic species invasions than monocultures (Hartman *et al.,* 2011). Finally, Tilman *et al.* (2006) identified that biofuels derived from diverse mixes required lower levels of agricultural inputs than their monoculture counterparts and thus could result in lower levels of fertilizer runoff and pollution.

Prairies must be managed regularly, most commonly by prescribed fire, grazing, or haying, to prevent encroachment by woody vegetation (Briggs et al., 2002). Biofuel harvesting incorporates mowing and haying, which offers an alternative scheme for managing prairie to that of burning or livestock grazing while resulting in some of the same conditions necessary for a healthy grassland (Fuhlendorf et al., 2006; Fargione *et al.*, 2009). Recent research has attempted to verify whether haying is a viable management option in terms of its effects on nesting grassland birds. Savannah sparrows (*Passerculus sandwichensis*) in Vermont, for example, exhibited no difference in nesting

success before and after a summer hay harvest, although the authors noted that this species characteristically maintains territories even after disturbances by harvest machinery (Zalik and Strong, 2008). Humbert (2010) showed that haying has a large detrimental impact on invertebrate populations; one might assume that the loss of invertebrate biomass resulting from the cutting and removal of vegetation would have a detrimental effect on nesting success for species that feed upon invertebrates. However, the absence of any difference in nesting success suggests that the sparrows were able to overcome this decrease in invertebrate biomass presumably by increasing the amount of time spent foraging (Zalik and Strong, 2008).

However, Perlut *et al.* (2008) found that abundance of savannah sparrows and bobolinks (*Dolichonyx oryzivorus*) in the northeast U.S. declined following haying treatments; the authors suggest that vulnerable populations in this region are probably not viable without any immigration from other populations. Furthermore, to increase the probability of persistence in the region given current harvesting methods, the authors suggested that a greater amount of fields should be harvested after songbirds had concluded nesting activities in mid- to late summer (Perlut *et al.*, 2008). Following harvests of switchgrass fields enrolled in the Conservation Reserve Program (CRP), abundances of sedge wren (*Cistothorus platensis*) and Henslow's sparrow (*Ammodramus henslowii*) were found to be significantly lower in harvested fields than in unharvested fields whereas abundances of grasshopper sparrow (*Ammodramus savannarum*) showed the opposite trend (Murray and Best, 2003; Roth *et al.*, 2005). These responses were reportedly due to differences in vegetation height and density following harvesting; Horn and Koford (2000) found that sedge wrens and other grassland species showed similar

avoidance of cut vegetation. Sedge wren, grasshopper sparrow, and Henslow's sparrow are all listed as Partners in Flight high priority species in several bird conservation regions in North America, and Henslow's sparrow is also listed as Near Threatened in the United States and Endangered in Canada (Rich *et al*., 2004).

Before biofuel harvest is widely adopted as a management option for native prairie grasslands, managers must consider the potential for these activities to be detrimental to grassland birds. To date, studies have been largely conducted on monoculture grasslands and it is uncertain to what extent these results will translate to native grassland systems. In our study we implemented a before-after control-impact (BACI) experimental design to determine the responses of grassland songbirds following harvesting of native biofuel vegetation (Stewart-Oaten *et al.*, 1986). BACI studies aid in the identification of environmental stresses (Smith, 2002), and we identified biofuel harvesting as such a potential stress. We predicted that species that showed declines in previous studies (e.g. sedge wren) would show similar declines following harvesting of native biofuel grassland vegetation, and species that previously showed increases would similarly show population increases (e.g. grasshopper sparrow). Our study was part of a larger field study conducted by the University of Minnesota's Cedar Creek Ecosystem Science Reserve to examine the potential for native, diverse grassland vegetation to be used for competitive and sustainable biofuel production (Williams *et al.*, 2013).

METHODS

Study area.—Our study area was comprised of a total of 59 plots in three regions of western Minnesota. The northwest region contained 16 plots, 4 of which were located on

Mentor State Wildlife Management Area (SWMA) in Polk County and the remaining 12 on private Conservation Reserve Program (CRP) fields in Polk and Red Lake counties. The west-central region contained 4 plots in Big Stone County located on Odden Waterfowl Production Area (WPA), 8 plots in Swift County located on Danvers SWMA and Artichoke Lake WPA, and 4 plots in Pope County located within Glacial Lakes State Park. The southwest region contained 13 plots in Jackson County located on Timber Lake SWMA, Heron Lake SWMA, and Heron Meadows SWMA; 6 plots in Nobles County located on Schweigert SWMA, Lone Tree SWMA, and West Graham SWMA; and 8 plots located on Talcot Lake SWMA in Cottonwood and Murray counties.

Plots consisted of 8 hectares of restored grasslands characterized by diverse mixes of native tallgrass vegetation that included big bluestem (*Andropogon gerardii*) and Indian grass (*Sorghastrum nutans*) as the dominant grasses. Prior to European settlement, these three regions were characterized by tallgrass prairie, but during the twentieth century nearly all of these grasslands were replaced by agricultural land (Samson and Knopf, 1994).

Harvesting.—Plots were harvested according to randomly assigned harvest percentages, including controls (not harvested), 50%, 75%, and 100% (fully harvested); once assigned a harvest percentage, each plot retained that particular treatment throughout the four years of study. The 50% and 75% harvest plots were also randomly assigned a harvest pattern categorized by either a block or strip harvest (Fig. 1). In October-November of each year from 2009 to 2012, tractors cut and baled the vegetation in plots so that roughly 15 cm of stubble remained (Jungers *et al*., 2013). For partial harvests (50% and 75%), the area of each plot that was harvested was rotated each year. Plots were no longer harvested

in the northwest region beginning in 2011 and in the west-central region beginning in 2012 as local processing facilities could no longer accommodate full production.

Bird surveys.—We conducted bird surveys each year from 2009 to 2013; the initial surveys were before impact, whereas surveys during 2010-2013 occurred after 1-4 consecutive years of biofuel harvest. Surveys occurred from mid-May to late June each year to coincide with the breeding period of most grassland birds in western Minnesota. The southwest region was surveyed first, followed by the west-central region and then the northwest region and finally a second round of surveys in the southwest region. The second round of surveys in the southwest region was conducted because we wanted to increase the likelihood of detecting late arriving species such as Dickcissel (*Spiza americana*). Two observers independently surveyed each plot per round such that each plot was not surveyed more than once per round by the same observer; however, in 2013 only one observer conducted surveys. The northwest region was not surveyed in 2013 as many of the plots had reverted back to agriculture following expiration of CRP contracts the previous year.

We used an area-based search method (Johnson and Igl, 1995) to survey birds in our plots (Appendix A). These surveys provide a greater probability of detection than traditional point counts and are easily conducted in flat, open terrain with short vegetation. Within each plot, we recorded the number of individuals of each species observed by either sight or sound. We only recorded birds that were perched in our plots at some point during the survey period, thereby excluding birds such as swallows that flew through the plots without stopping. Surveys began no sooner than 30 minutes after

sunrise and culminated no later than noon each day. Surveys were not conducted during high winds (>45km/hour) or persistent rain.

Statistical analyses.—For the species that were encountered in frequencies high enough to allow for abundance analyses (at least 150 individuals counted and present in at least 10% of our plots over the 5 years of study), we fit an intercept-only model using generalized linear models in R (The R Foundation for Statistical Computing, 2012). This allowed us to determine the statistical distribution (e.g. Poisson, zero-inflated Poisson, negative binomial, or zero-inflated negative binomial) that best fit our count data according to Akaike's Information Criterion corrected for small sample size (AIC_c).

We created 4 *a priori* models to predict species richness (defined as the total number of bird species per plot) and abundance and ranked these models according to AIC_c (Cunningham and Johnson, 2006). The first model included year as a linear covariate and percent harvest as an interaction with year; by including the effects of year in our a priori models of abundance, we recognized that bird abundance may exhibit linear trends. The second contained these same 2 variables, but included percent harvest as an additive effect and not as an interaction with year. The third and fourth models were identical to the first and second, respectively, except that year was treated as a before-after control-impact covariate (with values of 0 and 1 only). Because data collected in 2009 were pre-harvest, a harvest effect should show up as a year-by-harvest interaction, and more specifically, as a divergence of bird abundances or species richness across harvest treatments after 2009.

Next we created 2 more models, again identical to the first 2 except that year was treated as a factor with 5 levels. We then ranked the entire set of 6 models and chose the

best supported model as the template with which to create an exploratory model by adding additional covariates (using backward selection) that may be important in predicting abundance of these species. These covariates included: harvest pattern, the percent of grassland and wetland within a 250m radius of each plot, study region (e.g. northwest, west-central, or southwest Minnesota), date, cloud cover, wind speed, and survey starting time.

Harvest pattern was another component of our experimental design, but we excluded this variable in our *a priori* models because it had little support in past studies (Murray and Best, 2003) and we reasoned that if percent harvest did not impact species richness or abundance, then harvest pattern was unlikely to have an effect. We included percent grassland and wetland within a 250m radius because many species of grassland birds are area sensitive and respond to the amount of grassland habitat at much larger scales than our study plots (Johnson and Igl, 2001; Ribic and Sample, 2001). We chose a radius of 250m as the appropriate buffer as it corresponded with the average territory size of the species in our study with the largest breeding territory (bobolink; Fletcher and Koford, 2003). We calculated percent cover using ArcGIS (Environmental Systems Research Institute, 2010) and land cover data layers from the Upper Midwest Gap Analysis Program (GAP) vector layers for Minnesota (U.S. Geological Survey, 2000). We identified region as another potentially important landscape-level factor. Similarly, we considered survey date, cloud cover, wind speed, and start time of survey as potentially important spatio-temporal variables.

RESULTS

Species abundance and percent harvest.—We observed 57 bird species in our plots throughout the 5 years of study. Of these, 11 were encountered in frequencies great enough to allow for abundance analyses (Table 1). Le Conte's sparrow (*Ammodramus leconteii*, northwest region) and dickcissel (*Spiza americana*, southwest region) were only observed in one of the three regions of our study, and therefore our analysis of these two species were restricted to a single region. Common grackles (*Quiscalus quiscula*) were not observed in the northwest region, so we omitted that region in our abundance analysis of this species.

Abundance patterns of 6 species changed in post-harvest years. In the 2009 pre-harvest surveys, sedge wrens, common yellowthroats (*Geothlypis trichas*), clay-colored sparrows (*Spizella pallida*), and swamp sparrows (*Melospiza georgiana*) were observed in similar abundances across the four treatments or greater abundances in plots designated for greater harvest percentages (Figures 2, 3, 4, and 6). In 2010 and 2011 following biomass harvest, these patterns shifted to greater abundances in the 0-50% harvest range than in the 75-100% range, with 2012 being the year of the greatest magnitude of difference between abundances in controls and full harvest plots. Abundances of grasshopper sparrows and common grackles were similarly distributed in the pre-harvest year of 2009 but with greater numbers in the 0-50% harvest range (Figures 5 and 7). These patterns shifted over the next three years to greater abundances in the 75-100% harvest range, with 2012 and 2013 being the years of greatest magnitude in differences between controls and full harvest plots for grasshopper sparrows and common grackles, respectively.

Sedge wren, common yellowthroat, clay-colored sparrow, savannah sparrow, grasshopper sparrow, swamp sparrow, common grackle, and species richness all showed a response to harvesting; the best supported model for each analysis included the interaction between year and percent harvest (Table 2). Of these species, sedge wren, common yellowthroat, clay-colored sparrow, and swamp sparrow showed declines in abundance with increasing percent harvest (i.e. more birds were observed in controls than in full harvest plots). Similarly, species richness declined with increasing percent harvest. Grasshopper sparrow and common grackle were the only species that increased in abundance with increasing percent harvest (i.e. more birds were observed in full harvest plots than in controls).

Exploratory model covariates.—Harvest pattern was included in the exploratory models of 7 species and for species richness (Table 3), but only the species richness model indicated a response to the experimental treatment, as it included the interaction between harvest pattern and year. Harvesting in a strip pattern caused a slight decline in species richness compared to harvesting in a block pattern.

For clay-colored and grasshopper sparrows, as the percentage of grassland increased within 250m of the plots, so did their respective abundances. The opposite was true for sedge wren, common yellowthroat, savannah sparrow, swamp sparrow, red-winged blackbird (*Agelaius phoeniceus*), and common grackle, as well as for species richness. The percent wetland covariate was included in the models of sedge wren, common yellowthroat, clay-colored sparrow, savannah sparrow, grasshopper sparrow, red-winged blackbird, and common grackle, and species richness. For sedge wren, clay-colored sparrow, grasshopper sparrow, red-winged blackbird, and common grackle,

abundances increased as the percentage of wetland within 250m of the plots increased; the same was true for species richness. For common yellowthroat and savannah sparrow, abundances decreased as the percentage of wetland increased.

Region was included in all models except that of grasshopper sparrow (excluding Le Conte's sparrow and dickcissel). Red-winged blackbirds and common grackles were most abundant in the southwest and least abundant in the northwest. Sedge wrens, common yellowthroats, swamp sparrows, and bobolinks were most abundant in the west-central and least abundant in the northwest. Clay-colored and savannah sparrows were most abundant in the northwest region and least abundant in the southwest. Species richness was greatest in the southwest region.

Survey date was included in the models of common yellowthroat, grasshopper sparrow, dickcissel, and bobolink. For grasshopper sparrow and dickcissel, abundance increased as the date increased; the opposite was true for bobolink. Increasing cloud cover resulted in a slight increase in abundance of sedge wrens and a slight decrease in abundance of grasshopper sparrows. Wind speed was included in the models of sedge wren, common yellowthroat, clay-colored sparrow, savannah sparrow, and Le Conte's sparrow. For all but Le Conte's sparrow, increasing wind speed resulted in a decrease in abundance. Finally, the start time of surveys was included in the models of common yellowthroat, grasshopper sparrow, Le Conte's sparrow, swamp sparrow, dickcissel, bobolink, red-winged blackbird, and species richness. For all these species, abundance decreased with increasing start time, and the same was true for species richness.

DISCUSSION

Bird abundance in relation to harvest intensity.—Of 6 species that were affected by the amount of vegetation harvested via the different treatments, sedge wren, common yellowthroat, clay-colored sparrow, and swamp sparrow exhibited decreases in abundance with increasing percent harvest. For sedge wrens, this result was expected as other studies have linked sedge wren abundance to height and density of grassland vegetation (Delisle and Savidge, 1997; Murray and Best, 2003). In our study, harvesting resulted in a decrease in both vegetation height and density the following spring compared to unharvested areas. Vegetation density has also been identified as a limiting factor in the abundance of common yellowthroats (Patterson and Best, 1996), which is consistent with the observed decrease in abundance of this species in our plots following harvest. The preference of all these species for taller and denser vegetation may be explained by greater concealment of nests from predators (Burhans and Thompson, 1998; Kerns *et al.*, 2010).

Vegetation characteristics were also important predictors of the abundance of clay-colored sparrows, swamp sparrows, and grasshopper sparrows in our plots. Although clay-colored sparrows tend to nest in dense shrubby vegetation, their optimal foraging habitat appears to be away from the nest in grasslands and pasturelands (Knapton, 1980; Munson, 1992; Kerns *et al.*, 2010). Zalik and Strong (2008) found that invertebrate biomass decreases up to 82% following haying, which may explain the preference of clay-colored sparrows for the unharvested areas in our study. Beadell *et al.* (2003) found a negative correlation between the abundance of swamp sparrows and grasses of short height in coastal marshes. Although the habitat in our study tended to be drier than that of Beadell *et al.* (2003), we did observe that swamp sparrows were less

common in harvested plots, which suggests a similar relationship. Grasshopper sparrows, in contrast, prefer vegetation of shorter height (Murray and Best, 2003; Roth *et al.*, 2005), which suggests the increase in abundance as the percent of harvested area increased in our plots. Whitmore (1981) noted that sparser vegetation may result in easier and more efficient foraging for grasshopper sparrows.

The increase in abundance of common grackles with increasing percent harvest is somewhat surprising. Murray and Best (2003) noted no statistical difference in abundance of this species between harvested and unharvested treatments, but did find more grackles in unharvested areas. In our study, the inclusion of the interaction between percent harvest and year in the common grackle model is probably due to the increased number of grackles observed in 2013, as we noticed only slight changes in grackle abundance with harvest percentages in previous years.

We noted that 5 of the 6 species that were affected by percent harvest exhibited the greatest differences in abundance between controls and full harvest plots in 2012, with lesser effects from harvest in other post-harvest years. Results in 2013 may have been driven by unseasonably cool weather that persisted well into May. Combined with late receding snow cover, this caused a delay in the growing season such that vegetation height, particularly in our harvested areas, was exceptionally short (only several inches tall) at the onset of our field season in mid-May. Bird presence in harvested areas appeared lower than in previous years, although for the most part their respective trends in abundance from controls to full harvest plots in 2013 were similar to those seen in 2012. The exception was common grackle, which appeared in its greatest abundance in

2013 and seemed to be selecting the areas of very short stubble, exhibiting a sort of lawn foraging behavior.

One possible explanation for weak effects in 2010 and 2011 is a delayed response to the effects of harvesting. Changes in abundance did appear to be occurring in both 2010 and 2011 for sedge wrens, common yellowthroats, clay-colored sparrows, and grasshopper sparrows; Figures 2, 3, 5, and 6 potentially indicate abundances in one or both of these years as "shifting" from 2009 to 2012. Perhaps after a successful nesting season in 2009, birds came back to the same plots in 2010 and found that areas that had in past years been suitable nesting habitat had now become less suitable. These birds could have been detected in 2010 in similar abundances to the previous year, even if they did not stay on the plot to nest or if subsequent nesting attempts were less successful.

Studies of breeding site fidelity in grassland songbirds may support this possible explanation, although some studies found a link between breeding success or habitat variables and site fidelity whereas others did not. The return rate for adult grasshopper sparrows to breeding sites in subsequent years has been as high as 57% in some areas and as low as 8.9% in others (Balent and Norment, 2003; Gill *et al.*, 2006; Jones *et al.*, 2007; Ingold *et al.*, 2010). Adult savannah sparrows return rates varied from 38.7% to 5.4% (Bédard and LaPointe, 1984; Jones *et al.*, 2007; Ingold *et al.*, 2010). Bollinger and Gavin (1989) found the return rates of successfully nesting adult male and female bobolinks to be 55% and 41%, respectively while those of unsuccessful bobolinks were considerably lower (19% for males and 9% for females). Small *et al*. (2012) found that 21% of adult male dickcissels returned as did 30% of adult females. In all studies where return rates of hatch-year individuals were reported, their return rates were considerably lower than

those of adults (Gill *et al.*, 2006; Jones *et al.*, 2007; Ingold *et al.*, 2010; Small *et al.*, 2012). Few of these studies examined possible factors that led to higher or lower site fidelity.

The lack of a response to the amount of grassland harvested in our plots shown by savannah sparrows, Le Conte's sparrows, dickcissels, bobolinks, and red-winged blackbirds suggests that these species are unaffected by increasing percent harvest. Our results are in contrast to other studies that have found that savannah sparrows, bobolinks, and dickcissels prefer harvested areas (Delisle and Savidge, 1997; Horn and Koford, 2000; Murray and Best, 2003; Roth *et al.*, 2005). However, these studies were conducted in fields from about 8 to over 121 hectares in size, suggesting that some species are responding at a much larger scale than our study.

We detected more dickcissels in our full harvest plots than in controls in 2010, but the baseline pattern of abundance of this species in our plots had not been established in the control year as no dickcissels were observed in 2009. Dickcissel abundance in Minnesota fluctuates greatly from year to year as it does in other regions (Delisle and Savidge, 1997; Temple, 2002), and without the baseline pattern in 2009 for comparison we determined that the difference in abundance in 2010 was not biologically relevant.

Species Richness—Although we observed differences in mean species richness across our treatments, the resulting change in richness was less than 2 species in all post-harvest years in our study, which suggests a minimal effect of percent harvest at the community level. Similarly, Horn and Koford (2000) found only marginally significant differences in species richness between idled and mowed CRP fields in North Dakota, equating to a difference of about 2 more species in non-harvested areas than in harvested ones. Roth et

al. (2005) estimated overall species richness to be similar in both non-harvested and harvested transects, but did note that richness of species preferring grass of short or medium height did significantly increase in harvested transects and that the difference for short-grass species was due to the positive response of grasshopper sparrows. Murray and Best (2003) detected no major changes in species richness in their plots following harvesting, and also noted that the loss of one species was offset by the gain of another due to preferences in vegetation height.

Given that 4 species in our study showed decreases in abundance while 2 showed a significant increase in abundance with increasing percent harvest, one might assume that species richness would then be weighted toward the negative responses and therefore show declines of a greater magnitude. However, if abundant species declined by 50% and were still present in all our plots, species richness would remain unaffected. Additionally, our estimates of species richness included all species observed in a given plot, and not just the species that were included in our abundance analyses. It is therefore entirely possible that other species showed preferences for one treatment over another, and this may have resulted in a more mitigated effect of percent harvest on species richness in our plots (Murray and Best, 2003).

Other factors influencing abundance and species richness—The lack of response to the pattern of harvest suggests that whether an area is harvested via a strip or block pattern had little impact on the abundance of these 11 species. There was however a negative year-by-pattern interaction included in the exploratory model of species richness, and thus at the community level block harvests may encourage a greater number of species than strip harvests. Murray and Best (2003) noted the location of birds within fields of

both harvested and unharvested vegetation (similar to our 50% and 75% harvest plots) and conducted within-plot comparisons between the 2 treatments. They found that sedge wrens, common yellowthroats, song sparrows (*Melospiza melodia*), and red-winged blackbirds preferred the unharvested strips while grasshopper sparrows preferred the harvested strips (Murray and Best, 2003). Similarly, other studies have correlated abundance to the percent of vegetation within plots, differentiating between types of vegetation such as forbs or legumes (Patterson and Best, 1996). We, however, restricted patch size to the plot level in our study and did not relate abundances to patches within plots. Rather, a bird was recorded as occurring in a plot regardless of where it was observed within that plot.

Abundance of clay-colored and grasshopper sparrows increased as the amount of grassland increased within 250m of out plots. Similarly, Ribic et al. (2009) found a positive correlation between densities of bobolinks and eastern meadowlarks (*Sturnella magna*) and the percentage of grassland within 200m of their study sites in southwestern Wisconsin. Densities and probability of occurrence of some species—including sedge wren, clay-colored sparrow, grasshopper sparrow, Le Conte's sparrow, and bobolink—have been found to be positively correlated with larger grassland patches (Helzer and Jelinski, 1999; Johnson and Igl, 2001). Davis et al. (2006) identified only 1 species that increased in density as patch size increased, but also documented that some species experienced greater nesting success with increasing patch size. Four of the species in our study that showed negative responses to an increasing amount of grassland in the vicinity were common yellowthroat, sedge wren, swamp sparrow, red-winged blackbird, and common grackle. Common yellowthroats, swamp sparrows, and red-winged blackbirds

are not grassland specialists; rather, they are able to thrive in more heterogeneous landscapes (Yasukawa and Searcy, 1995; Mowbray, 1997; Guzy and Ritchison, 1999). Additionally, sedge wrens utilize a variety of wetland habitats in addition to grasslands (Herkert *et al.*, 2001), and without a wetland component more grassland might not offer optimal habitat for the wrens.

Bobolinks are an obligate grassland species, but studies have found that they do not avoid edges of agricultural fields as strongly as they avoid other edges (Fletcher and Koford, 2003; Bollinger and Gavin, 2004). This might explain why percentage of grassland within 250m was not important, as the dominant landscape surrounding the vast majority of our plots was agricultural fields. Winter *et al.* (2006) found that as patch size increased, densities of both savannah sparrows and bobolinks responded variably among 3 different regions, although the density of savannah sparrows increased with patch size in 2 of 3 regions.

The correlation between abundances and the amount of wetland within 250m of our plots proved rather interesting. We had initially hypothesized that wetland abundance would be positively correlated with abundances of common yellowthroat, swamp sparrow, and red-winged blackbird (Greenberg, 1988; Kim *et al.*, 2008). Although red-winged blackbird abundance increased as the amount of wetland increased, the opposite was true for common yellowthroat, and the swamp sparrow exploratory model did not include the wetland covariate as a predictor. These results are difficult to explain, although we did not distinguish between different types of wetlands within the GAP land cover. Thus it is possible that common yellowthroats and swamp sparrows were selecting against a certain type of wetland in proximity.

Species richness and abundances of sedge wren and grasshopper sparrow also increased as the amount of wetland increased within 250m. Wet conditions within grasslands have been found to attract a greater diversity of species due to more heterogeneous habitat (Kim *et al*., 2008). Sedge wrens tend to be associated with wetter grasslands (Herkert *et al.*, 2001), and perhaps that is what they were selecting in our study. Kim *et al*. (2008) suggested that available moisture in prairies may play an important role for grassland birds during times of drought. For at least two years in our study, western Minnesota experienced severe drought conditions, which may in part explain why an obligate grassland species like grasshopper sparrow exhibited increases in abundance with a greater amount of wetland in the vicinity.

Region was included in the models for 7 species and for species richness, suggesting that within western Minnesota there is regional variation at both the species and community levels. While these regional differences may represent actual regional differences in abundance, it is also possible that they may be a function of more local factors such as vegetation type or prior land use (e.g. CRP fields versus SWMAs) or even timing of breeding, since surveys were not conducted concurrently at all sites. As we had already excluded the region covariate from the models of Le Conte's sparrow and dickcissel, only for grasshopper sparrow was region not included as a predictor in the best supported model. This species is known to fluctuate in terms of its occupied range from year to year and is able to colonize new sites fairly quickly (Smith, 1963; Patterson and Best, 1996), thus this finding is not unexpected.

Of the spatio-temporal variables we included in our exploratory models, survey date, wind speed, and start time of surveys appeared to be more important than cloud

cover in that they were included in the models of more species. These variables probably reflect variation in detection probability (and perhaps migrational chronology in the case of survey date) rather than true variation in abundance, since it is unlikely that true abundance varied with wind speed, cloud cover, or time of day. Survey date was clearly an important predictor of dickcissel abundance, as in most years of our study this species was not present in southwest Minnesota until early June. Although this covariate was also included in the models of common yellowthroat, grasshopper sparrow, and bobolink, its effect on abundance was very small regardless of whether it was positive (grasshopper sparrow) or negative (common yellowthroat and bobolink). Abundances of sedge wren, common yellowthroat, clay-colored and savannah sparrows decreased with increasing wind speed, while abundance of Le Conte's sparrow increased. While it is intuitive that increased wind speed should hamper detection of most songbirds, it is difficult to explain why we tended to find more Le Conte's sparrows in windy conditions, or why wind speed apparently did not affect our detection of grasshopper sparrows. Start time of surveys was important in predicting abundances of six species and species richness. This again is intuitive as the singing rate of most songbirds tends to decrease as midday approaches, and indeed all abundances decreased as start time increased. Cloud cover was only included in the models of common yellowthroat and grasshopper, but its effects on the abundances of both species were minimal.

Finally, because two observers surveyed independently each year, and at least one of the observers was a different individual in each year, it is possible that observer error affected our measure of abundance for certain species, especially those that are characteristically difficult to detect by either sight or sound (e.g. grasshopper sparrow;

Diefenbach *et al.*, 2003). Similarly, Henslow's sparrows may have been more prevalent in some areas than we observed, but due to the difficulty in both hearing and seeing this species its abundance may have been underestimated.

Baseline patterns of abundance.—This study underscores the importance of longer-term studies as well as establishing baseline trends in relative abundance. In some years, a given species might be present in abundances too low to detect differences among treatments, and thus the chances of detecting significant responses to treatments increases with more years of study. Additionally, it cannot be assumed that a control year will result in a species being detected uniformly across the landscape, and thus determining those baseline patterns of abundance is paramount to understanding actual changes in abundance or species richness, particularly for grassland species that are present in low densities in a given year.

Management implications.—At the community level, harvesting native grasslands for biofuel production had little impact on grassland songbirds, which suggests that this is a viable management option for grassland songbirds. As CRP contracts begin to expire and grasslands revert back to row crops such as corn and soybeans, it may be necessary to mitigate the reduction in grassland bird habitat by encouraging additional lands set aside in order to sustain current populations of species that have benefited from the program (Johnson and Igl, 1995; McCoy *et al.*, 1999; Fargione *et al.*, 2009). Prior to harvesting native grasslands, however, it is important to identify what species are present and in the greatest abundances. If management plans include sedge wren as a species of concern, then greater percentages of grassland should be left unharvested. Conversely, if land managers are trying to provide optimal habitat for grasshopper sparrows, then greater

percentages can be harvested. We found little effect of the pattern of harvesting on any species, thus native grasslands can be harvested in either a block or strip pattern depending on the landscape. For example, if a large portion of a field contains rocks or woody vegetation, then the vegetation in that area could be left standing while the rest is harvested in a block pattern. For the harvesting machinery it is probably more efficient to harvest in a strip pattern, and this may be the optimal method if the landscape is more uniform and contains mostly herbaceous vegetation.

TABLE 1.—Eleven most abundant grassland bird species detected in experimental native grassland biofuel plots in western Minnesota, 2009-2013

Bird species	Individuals observed	Proportion of plots containing species
Bobolink (*Dolichonyx oryzivorus*)	2625	0.75
Red-winged blackbird (*Agelaius phoeniceus*)	1533	0.49
Savannah sparrow (*Passerculus sandwichensis*)	1197	0.52
Common yellowthroat (*Geothlypis trichas*)	960	0.51
Sedge wren (*Cistothorus platensis*)	814	0.40
Grasshopper sparrow (*Ammodramus savannarum*)	604	0.37
Clay-colored sparrow (*Spizella pallida*)	415	0.27
Swamp sparrow (*Melospiza georgiana*)	203	0.14
Dickcissel (*Spiza americana*)	201	0.13[a]
Common grackle (*Quiscalus quiscula*)	189	0.12[b]
Le Conte's sparrow (*Ammodramus leconteii*)	84	0.36[c]

[a] Only plots in the southwest region were included in analysis

[b] Only plots in the southwest and west-central regions were included in analysis

[c] Only plots in the northwest region were included in analysis

TABLE 2.—*A priori* models of species richness and abundance of 11 grassland songbird species in experimental native grassland biofuel plots in western Minnesota, 2009-2013. Models are ranked according to AIC_c, and the best supported model for each species and species richness is highlighted in bold font

	Year[a]*Harvest	Year+Harvest	Year(BACI)[b]*Harvest	Year(BACI)+Harvest
Sedge wren	**1840.370**	1860.441	1844.847	1855.966
Common yellowthroat	**2097.968**	2106.575	2118.107	2121.288
Clay-colored sparrow	**1297.819**	1298.830	1306.371	1307.941
Savannah sparrow	2335.123	**2332.721**	2342.049	2338.554
Grasshopper sparrow	**1666.772**	1671.507	1679.758	1679.177
Le Conte's sparrow	247.954	244.083	237.406	**233.994**
Swamp sparrow	739.491	746.746	**731.436**	745.849
Dickcissel	657.016	**655.852**	656.532	656.532
Bobolink	3254.102	**3252.127**	3263.248	3260.476
Red-winged blackbird	2473.992	**2470.028**	2479.603	2476.359
Common grackle	**656.718**	658.268	659.508	657.778
Species richness	**2779.441**	2785.559	2783.919	2786.457

[a] Year as a continuous covariate; 2009=0, 2010=1, etc.

[b] Year as a before-after control-impact covariate; pre-harvest=0, post-harvest=1

TABLE 3.—Parameter estimates (β) and standard errors (SE) in best supported models of species richness and abundance for 11 grassland songbird species in experimental native grassland biofuel plots in western Minnesota, 2009-2013

	Sedge wren		Common yellowthroat		Clay-colored sparrow	
Parameter	β	SE	β	SE	β	SE
Intercept	-0.125	0.344	4.988	0.973	-0.046	0.403
Year[a]			-0.162	0.076		
YearBACI[b]						
YearF1[c]	-0.638	0.398			0.576	0.459
YearF2	-0.821	0.370			0.164	0.458
YearF3	0.827	0.347			-0.053	0.496
YearF4	-0.849	0.501			-0.066	0.667
%Harvest	-0.325	0.365	-0.296	0.237	1.105	0.445
Year:%Harvest			-0.298	0.111		
YearBACI:%Harvest						
YearF1:%Harvest	-0.463	0.557			-1.059	0.617
YearF2:%Harvest	-0.612	0.523			-1.657	0.630
YearF3:%Harvest	-2.418	0.505			-2.065	0.701
YearF4:%Harvest	-1.291	0.758			-0.631	0.900
PatternStrip[d]	-	-	-0.117	0.118	-	-
Year:PatternStrip	-	-	-	-	-	-
%Grassland	-1.252	0.318	-0.830	0.309	0.919	0.423
%Wetland	4.281	0.875	-2.573	0.924	2.077	0.887
RegionWC	1.967	0.259	0.100	0.261	-0.391	0.276
RegionSW	1.643	0.250	0.052	0.281	-1.785	0.262
Date	-	-	-0.020	0.006	-	-
Cloud	-	-	0.033	0.045	-	-
Wind	-0.344	0.094	-0.132	0.074	-0.354	0.118
StartTime	-	-	-	-	-	-
Dispersion parameter	0.689	0.085	0.749	0.201	0.640	0.105

[a] Year as a continuous covariate

[b] Year as a before-after control-impact covariate

[c] YearF1-F4: Year as a factor

[d] Block harvest=0, strip harvest=1

TABLE 3.—Continued

	Savannah sparrow		Grasshopper sparrow		Le Conte's sparrow	
Parameter	β	SE	β	SE	β	SE
Intercept	2.021	0.149	0.770	1.058	0.207	0.967
Year[a]	-0.048	0.040	-0.435	0.102	0.525	0.228
YearBACI[b]						
YearF1[c]						
YearF2						
YearF3						
YearF4						
%Harvest	-0.187	0.130	-0.573	0.272	0.012	0.356
Year:%Harvest			0.466	0.131		
YearBACI:%Harvest						
YearF1:%Harvest						
YearF2:%Harvest						
YearF3:%Harvest						
YearF4:%Harvest						
PatternStrip[d]	-0.219	0.110	0.508	0.124	-0.031	0.357
Year:PatternStrip	-	-	-	-	-	-
%Grassland	-0.322	0.336	0.770	0.208	-	-
%Wetland	-0.641	0.561	0.606	0.649	-	-
RegionWC	-0.595	0.166	-	-		
RegionSW	-1.299	0.162	-	-		
Date	-	-	0.009	0.006	-	-
Cloud	-	-	-0.060	0.047	-	-
Wind	-0.071	0.058	-	-	1.124	0.443
StartTime	-	-	-0.155	0.037	-0.183	0.085
Dispersion parameter	1.568	0.264	2.247	0.695	1.000	-

[a] Year as a continuous covariate

[b] Year as a before-after control-impact covariate

[c] YearF1-F4: Year as a factor

[d] Block harvest=0, strip harvest=1

TABLE 3.—Continued

	Swamp sparrow		Dickcissel		Bobolink	
Parameter	β	SE	β	SE	β	SE
Intercept	-2.398	1.232	-16.70	2.455	2.805	0.703
Year[a]			0.422	0.134		
YearBACI[b]	1.259	0.517				
YearF1[c]					-0.268	0.143
YearF2					-0.290	0.126
YearF3					-0.383	0.130
YearF4					-0.722	0.152
%Harvest	0.017	0.617	-0.710	0.377	-0.063	0.111
Year:%Harvest						
YearBACI:%Harvest	-2.715	0.709				
YearF1:%Harvest					-	-
YearF2:%Harvest					-	-
YearF3:%Harvest					-	-
YearF4:%Harvest					-	-
PatternStrip[d]	-0.656	0.263	-	-	0.257	0.090
Year:PatternStrip			-	-	-	-
%Grassland	-4.600	0.669	-	-	-	-
%Wetland			-	-	-	-
RegionWC	5.184	1.074			0.699	0.142
RegionSW	4.998	1.034			0.289	0.143
Date			0.111	0.014	-0.008	0.004
Cloud			-	-	-	-
Wind			-	-	-	-
StartTime	-0.213	0.072	-0.257	0.082	-0.011	0.027
Dispersion parameter	0.581	0.138	0.414	0.080	0.686	0.135

[a] Year as a continuous covariate

[b] Year as a before-after control-impact covariate

[c] YearF1-F4: Year as a factor

[d] Block harvest=0, strip harvest=1

TABLE 3.—Continued

	Red-winged blackbird		Common grackle		Species richness	
Parameter	β	SE	β	SE	β	SE
Intercept	-0.548	0.882	-2.824	0.871	1.615	0.134
Year[a]					0.033	0.029
YearBACI[b]						
YearF1[c]	-0.125	0.198	-0.014	0.522		
YearF2	-0.054	0.189	-0.873	0.489		
YearF3	-0.589	0.195	-1.340	0.509		
YearF4	-0.568	0.231	-0.082	0.550		
%Harvest	0.112	0.217	0.991	0.513	-0.047	0.091
Year:%Harvest					-0.101	0.040
YearBACI:%Harvest						
YearF1:%Harvest			-	-		
YearF2:%Harvest			-	-		
YearF3:%Harvest			-	-		
YearF4:%Harvest			-	-		
PatternStrip[d]	-	-	-0.654	0.388	0.025	0.072
Year:PatternStrip	-	-	-	-	-0.053	0.032
%Grassland	-0.518	0.450	-1.478	0.749	-0.480	0.088
%Wetland	3.914	2.304	9.890	2.909	0.900	0.255
RegionWC	1.636	0.516			0.555	0.073
RegionSW	2.533	0.845	2.101	0.633	0.606	0.068
Date	-	-	-	-	-	-
Cloud	-	-	-	-	-	-
Wind	-	-	-	-	-	-
StartTime	-0.044	0.043	-	-	-0.044	0.012
Dispersion parameter	-0.153	0.130	0.110	0.020	1.000	-

[a] Year as a continuous covariate

[b] Year as a before-after control-impact covariate

[c] YearF1-F4: Year as a factor

[d] Block harvest=0, strip harvest=1

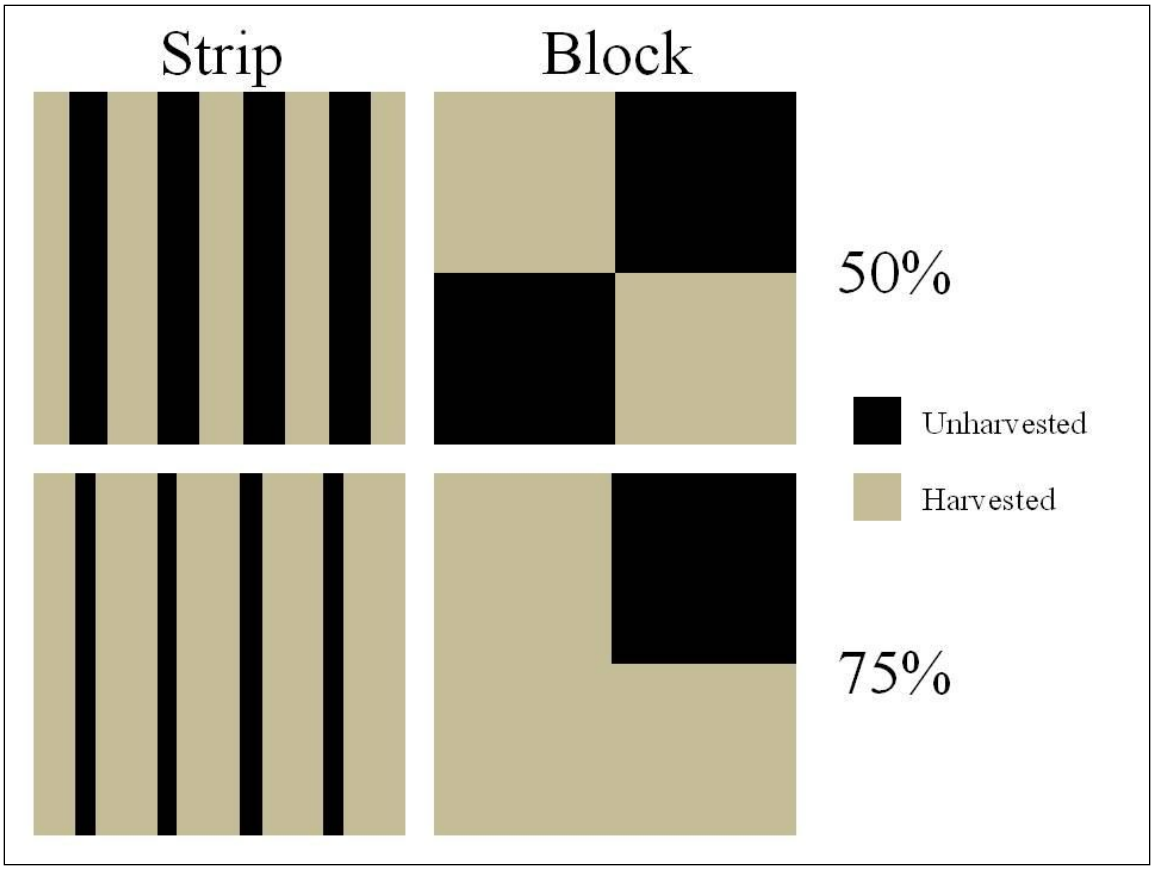

FIG. 1.— Harvesting differences between 50% and 75% harvest plot treatments in experimental native grassland biofuel plots in western Minnesota, 2009-2013

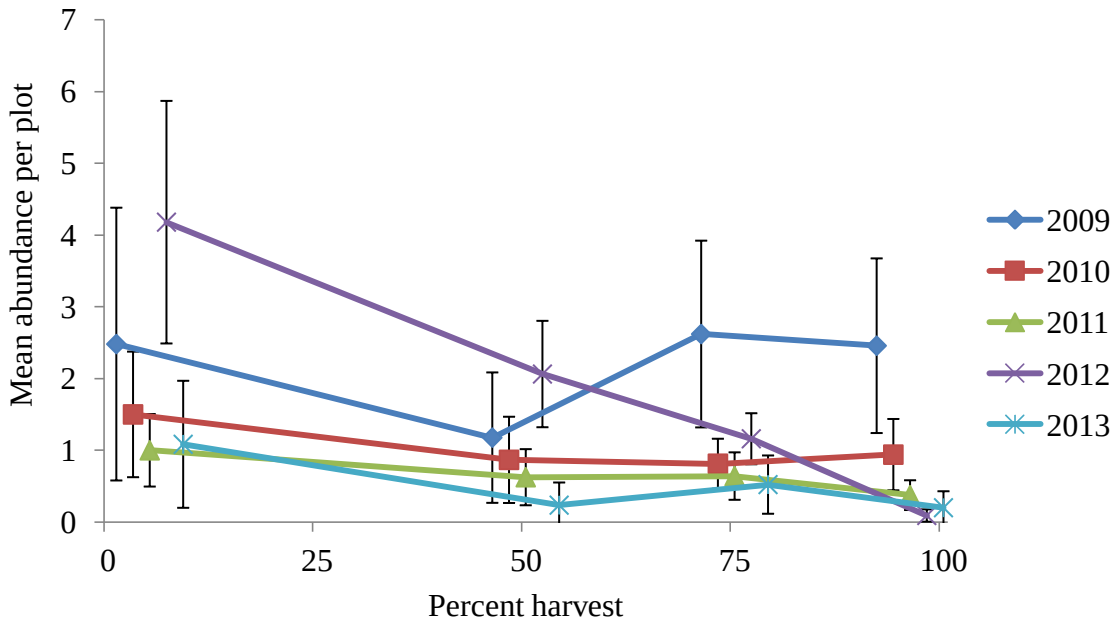

FIG. 2.—Mean abundance of sedge wrens per plot ± 95% confidence intervals in relation to percent harvest of experimental native grassland biofuel plots in western Minnesota, 2009-2013

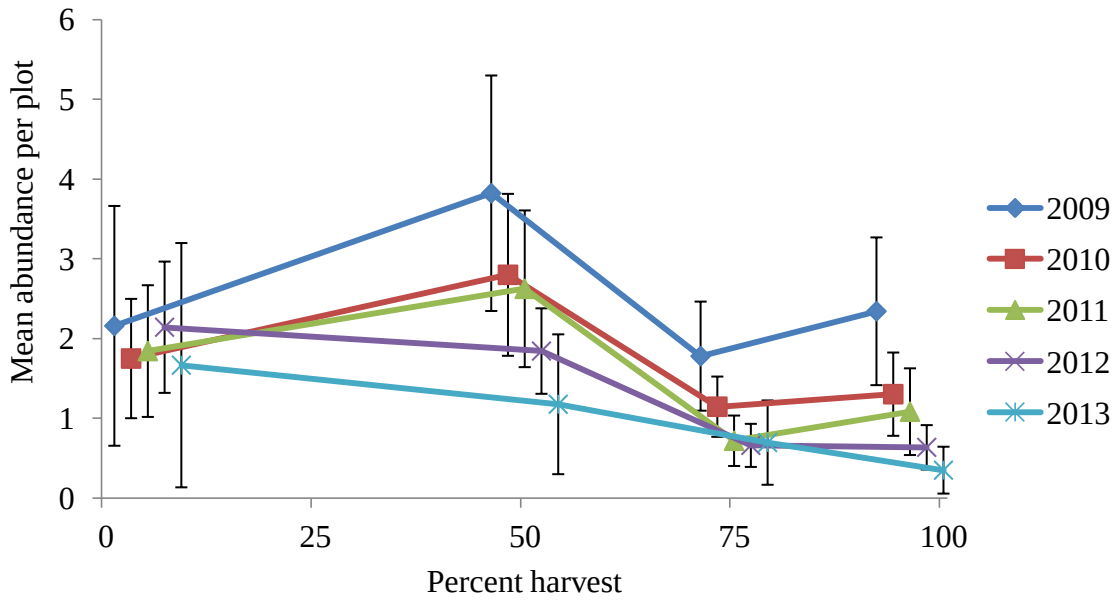

FIG. 3.—Mean abundance of common yellowthroats per plot ± 95% confidence intervals in relation to percent harvest of experimental native grassland biofuel plots in western Minnesota, 2009-2013

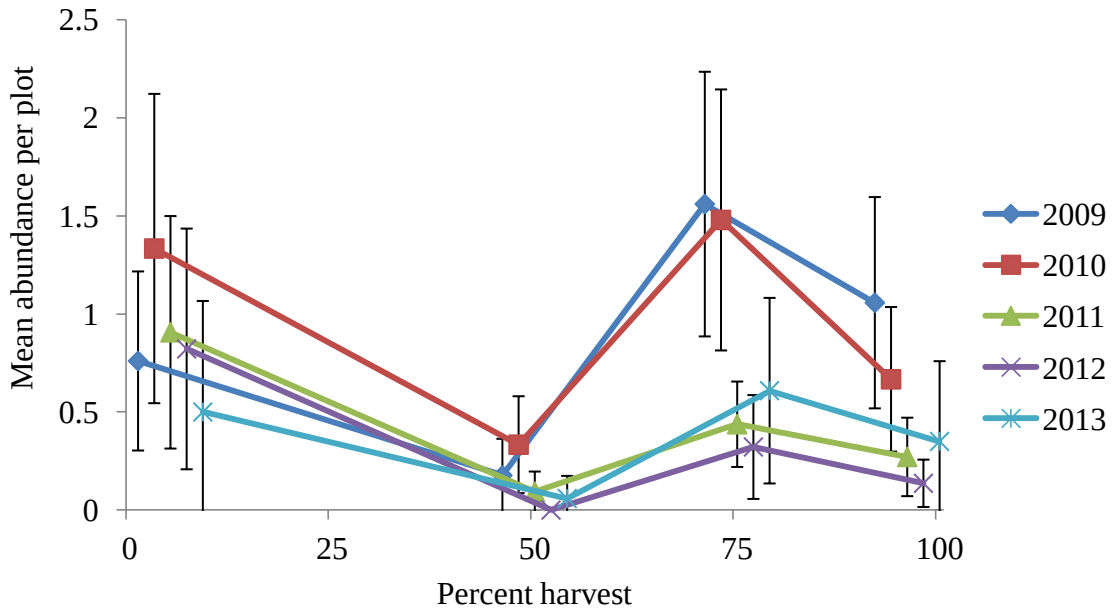

FIG. 4.—Mean abundance of clay-colored sparrows per plot ± 95% confidence intervals in relation to percent harvest of experimental native grassland biofuel plots in western Minnesota, 2009-2013

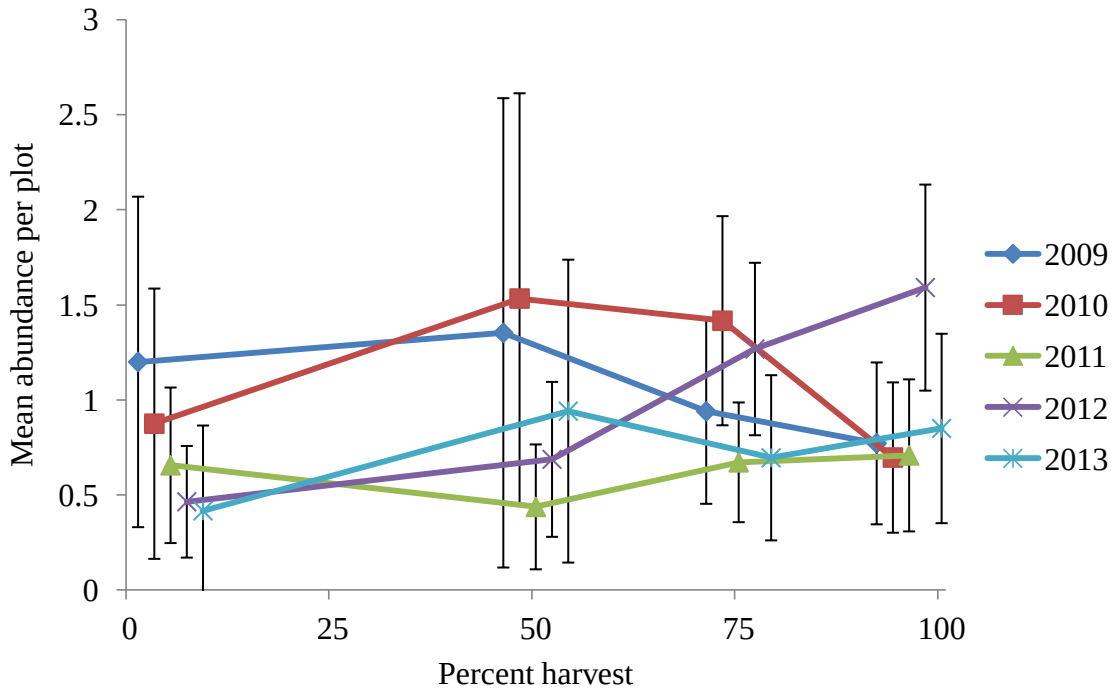

FIG. 5.—Mean abundance of grasshopper sparrows per plot ± 95% confidence intervals in relation to percent harvest of experimental native grassland biofuel plots in western Minnesota, 2009-2013

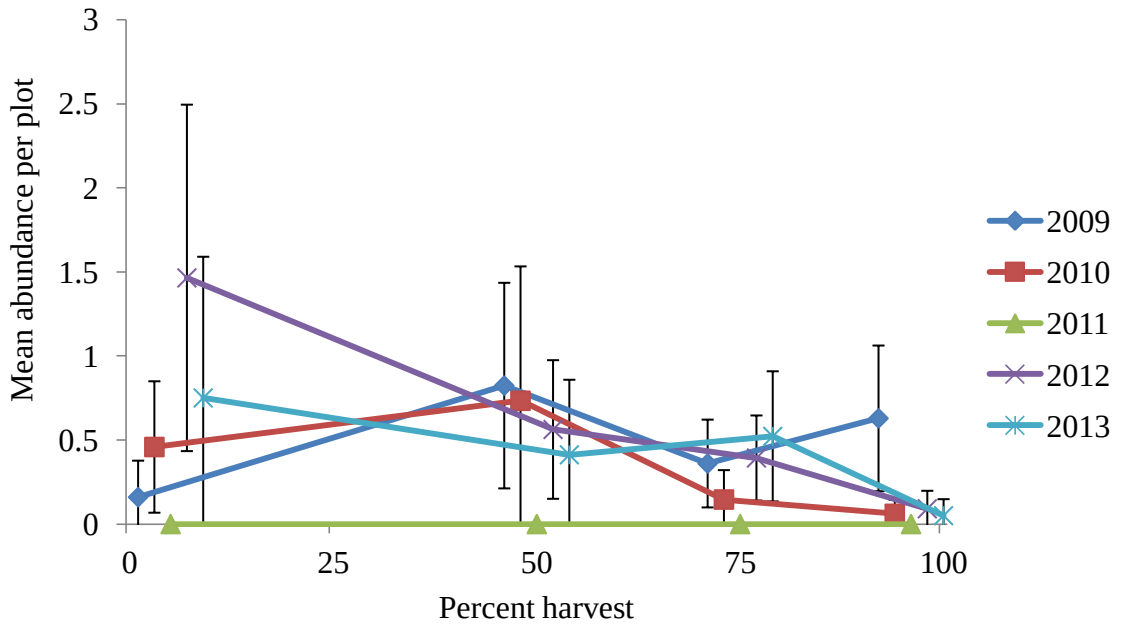

FIG. 6.—Mean abundance of swamp sparrows per plot ± 95% confidence intervals in relation to percent harvest of experimental native grassland biofuel plots in western Minnesota, 2009-2013

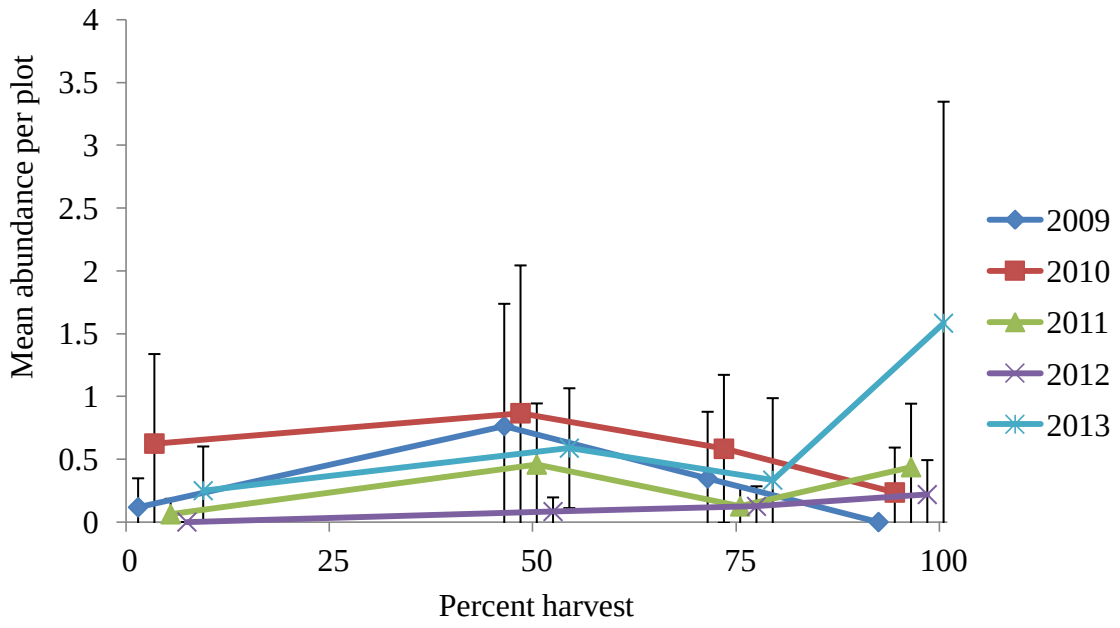

FIG. 7.—Mean abundance of common grackles per plot ± 95% confidence intervals in relation to percent harvest of experimental native grassland biofuel plots in western Minnesota, 2009-2013

CHAPTER 2: SHORT-TERM EFFECTS OF NATIVE GRASSLAND BIOFUEL HARVESTS ON SMALL MAMMALS IN WESTERN MINNESOTA

Harvesting native grassland vegetation for biofuels has great potential to offer a source of renewable energy for human consumption as well as habitat for native wildlife (Tilman et al. 2006, Fletcher et al. 2011, Hartman et al. 2011). Small mammals such as voles and shrews, however, have declined in density and abundance following mowing of prairie vegetation (Semere and Slater 2007, Washburn and Seamans 2007). These declines have been associated with decreases in ground cover and vegetation height, which may be detrimental to small mammals by making them become more exposed and visible to predators (Semere and Slater 2007).

Declines in small mammal populations are potentially worrisome. In addition to their roles as ecosystem engineers that promote greater biodiversity (Questad and Foster 2007), small mammals also constitute an abundant source of food for a variety of predators. Bank voles (*Clethrionomys glareolus*) are preyed upon by at least 17 species in Europe, including raptors, foxes, and other small mammals such as weasels (Jędrzejewski et al. 1993). In North America, small mammals comprise the majority of diets of several hawk species—including northern harrier (*Circus cyaneus*), red-tailed hawk (*Buteo jamaicensis*), rough-legged hawk (*Buteo lagopus*), and ferruginous hawk (*Buteo regalis*)—as well as those of mammalian and reptilian predators (Sherrod 1978). A decline in small mammal populations, then, could cause detriment to populations of their predators, many of which are apex predators in grassland ecosystems.

Studies in North America and Europe have identified voles of the genus *Microtus* as the most abundant small mammals in grassland habitats (Snyder and Best 1998, Kaufman et al. 2000, Semere and Slater 2007), and thus the literature studying their

ecology is fairly extensive. In relation to grassland harvests, several studies have documented *Microtus* declines following mowing (Lemen and Clausen 1984, Seitman et al. 1994, Edge et al. 1995, Slade and Crain 2006) and grazing (Rosenstock 1996, Wheeler 2008). Studies have also identified the importance of vegetation cover density (Birney et al. 1976, Tattersall et al. 2000) as well as fragmentation and patch size (Bowers et al. 1996, Pasitschniak and Messier 1998, Cook et al. 2004) to *Microtus* populations. In terms of biofuel harvesting, haying methods cause both fragmentation and a removal of vegetation at least on a local scale as harvested vegetation leaves behind matrices of shorter, less dense stubble that might not be suitable habitat for microtines. Before landowners adopt large scale harvesting of native grassland vegetation, it is thus imperative that these activities' potential for detriment to *Microtus* and other small mammals is identified.

Our study is part of a larger study on the potential for native grassland vegetation for use in competitive and sustainable biofuel production conducted by the University of Minnesota's Cedar Creek Ecosystem (Williams et al. 2013). In our study we implemented a before-after control-impact (BACI) design to determine the responses of small mammals to experimental harvest of grassland vegetation (Stewart-Oaten et al. 1986). We predicted that *Microtus* would show population declines similar to those in previous studies, and that these declines would be cumulative as ground cover continued to be reduced in subsequent years.

STUDY AREA

Small mammals were surveyed in three regions of western Minnesota: southwest, west-central, and northwest. The northwest region consisted of 16 plots near Crookston,

Minnesota in Polk and Red Lake counties. The west-central region consisted of 16 plots near Morris, Minnesota in Big Stone, Pope, and Swift counties. The southwest region consisted of 27 plots near Windom, Minnesota in Cottonwood, Jackson, Murray, and Nobles counties; extra funding was available in the southwest region to consider extra treatments. Plots were roughly 8 ha in size and located on lands characterized by various types of ownership and management, including state managed wildlife management areas (WMAs), a state park, federally managed waterfowl production areas (WPAs), and privately owned lands in the Conservation Reserve Program (CRP). Study sites were chosen for their logistical consideration, including landowner willingness to allow long-term biomass harvest, in addition to meeting the criteria of 8 contiguous hectares of restored native grassland. For further description, see Williams et al. (2013).

The western third of Minnesota was predominantly characterized by tall-grass prairie pre-European settlement and is now largely comprised of agricultural lands (Samson and Knopf 1994). Our plots consisted of restored prairie dominated by big bluestem (*Andropogon gerardii*) and Indian grass (*Sorghastrum nutans*) in addition to mixed forbs. Plots were primarily adjacent to agricultural land, but smaller amounts of wetlands and other grasslands were also in the vicinity.

METHODS

Field Methods

We conducted small mammal surveys from mid-August to mid-September each year from 2009 to 2012. We visited each of our three study regions once per year during the first three years of the study, but only the southwest and west-central regions in the final year. We surveyed all plots in a region before moving on to the next region. Within each

plot we set and marked 49 Sherman live traps with numbered flags in a 7 by 7 grid centered in the center of the plot, with 15 m between traps (Slade and Crain 2006). We set traps in the afternoon of the first day and checked and closed them the following morning; we repeated this process for a total of 3 nights at each plot in 2009 and 4 nights at each plot in 2010, 2011, and 2012. We baited traps with a mixture of oats and peanut butter, along with 4 full-sized cotton balls for bedding and insulation. Captured individuals were identified to species prior to release, but due to difficulties discriminating *Microtus ochrogaster* from *Microtus pennsylvanicus* and *Peromyscus leucopus* from *Peromyscus maniculatus*, we combined these species into the *Microtus* and *Peromyscus* genera, respectively. Captures individuals were marked on their bellies using a colored marker and recaptures were noted. We handled animals in accordance with guidelines for mammal collection approved by the American Society of Mammalogists (Animal Care and Use Committee 1998). Additionally, we followed protocols for the study of free ranging wildlife as delineated by the Institutional Animal Use and Care Committee (IACUC) for the University of Minnesota.

In October-November of each year, plots were harvested using a self-propelled windrower with an attached mounted disc cutter (Jungers et al. 2013). Plot treatments included a full harvest (100%), a control (0%), rotating block harvests of 50% and 75%, and rotating strip harvests of 50% and 75% (Fig. 1). Prior to the first harvest, each plot was randomly assigned a treatment that included one of these harvest percentages, and each plot retained this same treatment throughout the entire study. If the plot was assigned a harvest percentage of either 50% or 75%, then it was also randomly assigned a harvest pattern (i.e. strip or block). Because we had fewer plots in the west-central and

northwest regions, 50% harvest was excluded as a treatment in those regions. Additionally, harvesting did not take place in the northwest in 2011 as local processing facilities could no longer accommodate full production.

Following the BACI model, the small mammal surveys we conducted in 2009 before the October-November harvest were surveyed in a pre-harvest or control year. The small mammals we surveyed in 2010, 2011, and 2012 were surveyed in post-harvest years (i.e. harvest took place the preceding October-November).

Statistical Analyses

We used total individuals captured per 4 nights per plot (3 nights in 2009) excluding recaptures as an index of relative abundance (Semere and Slater 2006), and we only counted recaptured individuals once per year. For species or genera that we captured frequently enough (at least 400 individuals in at least 40% of our plots), we analyzed relative abundance using generalized linear models in R (The R Foundation for Statistical Computing, 2012). We used intercept-only models to identify the statistical distribution (e.g. Poisson, zero-inflated Poisson, negative binomial, or zero-inflated negative binomial) that best fit our count data according to Akaike's Information Criterion corrected for small sample size (AIC_c).

We created 12 a prior models to predict species richness (defined as total number of small mammal species/genera per plot) and abundance and ranked them according to AIC_c (Cunningham and Johnson, 2006). The first model included year and percent harvest as continuous covariates, along with a year-by-percent harvest interaction. The second model contained these same two variables, but without the interaction effect. Because data collected in 2009 were pre-harvest, a harvest effect should show up as a

year-by-harvest interaction, and more specifically, as a divergence of small mammal abundances or species richness across harvest treatments after 2009. Alternatively, the inclusion of percent harvest as an additive effect (i.e., without an interaction) would suggest that abundances varied among the plots prior to the first harvest, and that these patterns of initial abundance remained unchanged throughout the study. The third and fourth models contained the experimental covariate for harvest pattern (i.e. strip or block) as an interaction with year and as an additive effect, respectively. Similar to percent harvest, an effect of harvest pattern should show up as a year-by-harvest pattern interaction. The fifth and sixth models included both harvest percentage and harvest pattern as interactions with year (model 5) and as separate additive effects (model 6). The remaining 6 models were identical to the first 6 except that year was treated as a before-after control-impact dummy covariate, with values of 0 for pre-harvest and 1 for post-harvest.

Using the best supported a priori model as a template, we included region (e.g. northwest, west-central, or southwest Minnesota) as a factor variable to determine if small mammal abundance and species richness varied among study regions.

For small mammals with fewer than 400 total captures, or for those captured on less than 40% of study plots, we used robust design occupancy modeling (MacKenzie et al. 2003) to examine patterns of species occurrence throughout the 4 years of our study. For each study plot, we created an encounter history with 15 encounter occasions (3 trapping nights in 2009, and 4 trapping nights in each year 2010-2012), where each encounter was either "1" (at least one individual was captured) or "0" (none captured) for each species/genera in each plot. Our field methods met the assumptions of robust

design occupancy in that there were open periods between primary sampling periods (years) where species could potentially disappear from or recolonize study plots and closed secondary sampling periods (concurrent trap nights) where presence or absence of a particular species in the study plot was unlikely to change. We modeled robust design occupancy using Program MARK (White and Burnham 1999) to estimate the probability that plots were occupied by each species during the first year of the study (ψ), the annual probability of an occupied site becoming locally extinct (ε), the annual probability of an unoccupied site being colonized (γ), and the probability of detecting a species during any given trapping night, given it was present (p). We created a priori models that included effects of year and species on occupancy and detection probabilities as well as the random effects of percent harvest and harvest pattern in addition to these covariates on extinction and colonization probabilities. We ranked these models according to AIC_c and chose the best supported model predicting local extinction and colonization probabilities.

RESULTS

We captured 4,443 small mammals were captured during our study, including 2,601 *Microtus* spp., 845 northern short-tailed shrews (*Blarina brevicauda*), 132 meadow jumping mice (*Zapus hudsonias*), 107 short-tailed weasels (*Mustela erminea*), 190 thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*), 153 masked shrews (*Sorex cinereus*), and 425 deer mice (*Peromyscus* spp.). We were able to perform abundance analyses for *Microtus*, northern short-tailed shrews, and deer mice due to their large sample sizes. We also caught 7 northern grasshopper mice (*Onychomys leucogaster*), 1 plains pocket mouse (*Perognathus flavescens*), 1 western harvest mouse (*Reithrodontomys megalotis*), and 1 house mouse (*Mus musculus*), but these individuals

were excluded from our analyses due to insufficient samples. Deer mice were not observed in our northwest region, and thus we excluded this region in our analysis of the abundance of this genus.

Abundance patterns of *Microtus* changed in post-harvest years, while those of northern short-tailed shrews and deer mice were more constant. In 2009, *Microtus* were captured in similar numbers across the four harvest percentages (Fig. 2). In the following years, this pattern shifted to more captures in our controls than in full harvest plots, with the final year of the study exhibiting the greatest magnitude of difference between controls and full harvest plots. Northern short-tailed shrews were more abundant in post-harvest years, but their pattern of abundance did not exhibit much change from 2009 (Fig. 3). Deer mice were captured in our plots in low numbers in the first three years of our study, and their abundance trends across the treatments appeared fairly constant (Fig. 4). In the final year of the study, however, deer mice were captured very frequently and exhibited an increase in abundance as the amount of biofuel harvest increased.

Table 1 shows the best supported a priori models of abundance and species richness as well as all models within 2 ΔAIC_c. *Microtus* was the only small mammal that responded to harvesting; as the amount of grassland harvested increased, *Microtus* abundance decreased. The best supported models for northern short-tailed shrews, deer mice, and species richness only included the additive effects of percent harvest. Year was included as a continuous covariate only for the *Microtus* model; the models of northern short-tailed shrews, deer mice, and species richness all included year as a BACI covariate. Harvest pattern was not included as an important predictor in any of the models. When region was added to the best supported a priori models for *Microtus* and

northern short-tailed shrews, the resulting models became the best supported models. *Microtus* were most abundant in the northwest and least abundant in the southwest, whereas the opposite was true for northern short-tailed shrews (Table 2).

The best supported robust design occupancy model included the effects of year, species, and percent harvest on local extinction and colonization probabilities in addition to the effects of year and species on occupancy and detection probabilities (Table 3). Table 4 shows the beta estimates of the effect of percent harvest on local extinction and colonization probabilities of each species in our plots as predicted by this model. Local extinction rates increased with increased harvest percentage for *Microtus*, short-tailed weasels, thirteen-lined ground squirrels, and masked shrews, while local extinction rates decreased for northern short-tailed shrews, meadow jumping mice, and deer mice. Colonization rates of *Microtus*, northern short-tailed shrews, and masked shrews decreased with increasing percent harvest, while colonization rates of meadow jumping mice, short-tailed weasels, thirteen-lined ground squirrels, and deer mice increased.

DISCUSSION

Our abundance analyses showed that only *Microtus* were affected by harvest percentage, declining with increasing harvest percentage. This decline is well supported in existing literature on *Microtus* responses to mowing or grazing of grassland vegetation (Lemen and Clausen 1984, Seitman et al. 1994, Edge et al. 1995, Rosenstock 1996, Slade and Crain 2006, Wheeler 2008). It is plausible that mowing resulted in a loss of suitable cover for *Microtus*, as refuge from predators has been identified as a limiting factor for their populations (Birney et al. 1976, Tattersall et al. 2000). Thus leaving some percent of

standing, unharvested vegetation in fields is likely paramount to producing greater numbers of *Microtus* in harvested grasslands.

Pasitschniak and Messier (1998) found that *Microtus* were more abundant along edges of grasslands rather than the interiors, although they noted that this relative abundance appeared to be related to seasonal changes in that abundance was greater in spring than in summer. Similarly, Bowers et al. (1996) determined that edges offered higher quality home range sites for *Microtus*. In our study, strip harvests created a larger amount of edge in relation to block harvests, but we found no evidence of edge effects on *Microtus* in our study as our best supported model of *Microtus* abundance did not include the harvest pattern covariate. Additionally, *Microtus* movements have been observed between patches following fragmentation; Cook et al. (2004) found that movements increased as patch size decreased. We did not survey small mammals at the patch scale, but patch scale movements suggest that *Microtus* are able to use the matrix between patches (Cook et al. 2004). This may explain why harvest pattern was not a significant predictor of *Microtus* abundance at the plot level.

The inclusion of year as a continuous covariate in our *Microtus* model suggests that the effects of harvest percentage are cumulative. This result was expected, as density of ground cover and litter can take several years to recover following mowing, particularly in years of drought (Lemen and Clausen 1984). *Microtus* populations are known to fluctuate with a cycle of 3 to 4 years, although authors have noted that these multiannual cycles are muted in grassland habitat (Getz and Hofmann 1999, Getz et al. 2001). As our study took place over 4 years, it is difficult to know if the cumulative effect of several years of harvesting was in part a function of the multiannual cycles of this

genus. However, we did note that total *Microtus* abundance in our plots increased in both 2010 and 2011; if the cycles did play a role in our study, it was probably small.

Abundance of deer mice in our plots clearly increased in 2012 as captures were approximately 4 times more frequent than in previous years. Deer mice also appeared to prefer harvested plots to unharvested ones in that year, although this preference in one year of the study was not large enough to add support for the year-by-percent harvest interaction in our best supported model; 2012 may have represented an uncommon irruption of this genus in western Minnesota (Drost and Fellers 1991). The previous three years showed no trends of either decreasing or increasing abundance, and in 2011 very few deer mice were caught in our plots. Some authors have identified multiannual cycles of density in deer mice populations, although others have noted mostly stable populations that vary within but not between years (Drost and Fellers 1991, Van Horne 1981). With 4 years of study, it is difficult to determine if a multiannual cycle was indeed exhibited by deer mice in our plots or instead the population boon in 2012 was the result of an irruption caused by some environmental factor.

Van Horne (1981) concluded that abnormally high densities of deer mice were a result of an early breeding season in a relatively low population of adults, and these conditions certainly would have been met in our study in 2012 as unusually warm spring temperatures throughout the Upper Midwest followed a year of low deer mouse populations in our plots. Regardless, the increasing abundance observed with increasing percent harvest in 2012 at first seems counterintuitive, since less cover translates to greater exposure to predators. However, Kaufman et al. (1988) proposed that the removal of ground litter allows for more grains and seeds—the preferred food of deer mice—to

become exposed on the bare ground, which provides for less energy-expensive foraging. Perhaps for deer mice the benefits of easier foraging outweigh the risks of greater chances of predation, and this may be especially true in years when this genus is more abundant.

Northern short-tailed shrew abundance was relatively constant in the 4 years of study. Getz and McGuire (2008) also observed that vegetation cover had no effect on northern short-tailed shrew movements or home range size.

Species richness also showed little evidence of changing trends, and any differences between years were likely related to temporal or environmental effects (Semere and Slater 2007). In contrast to our results, Semere and Slater (2007) found that diversity of small mammals increased slightly from cropped areas to field margins. This suggests that areas of greater harvest intensity (e.g. our full harvest plots) should have fewer species than areas with more vegetation (e.g. our controls). While we did observe lower richness in full harvest plots than in control plots in all 4 years of our study, at most one species was lost when the amount of plot harvested was 100%. Thus the amount of grassland harvested probably plays only a small role in determining species richness within small mammal communities.

In our occupancy analysis, we observed annual variation in local extinction and colonization probabilities, indicating that population persistence varied from year to year (MacKenzie et al. 2003). Similarly, we observed species effects, because different species are unlikely to be present in equal proportions over a given number of plots; rather, they are distributed nonrandomly (Kaufman et al. 2000). As in our models of

abundance and species richness, harvest pattern (i.e., strip or block) had no effect on local extinction or colonization probabilities for any species.

Only the local extinction and colonization rates of *Microtus* and masked shrews showed opposite trends as local extinction increased and colonization decreased. This suggests that these two small mammals were negatively affected by the amount of grassland harvested. We saw a similar effect of harvest percentage in our *Microtus* abundance analysis, thus it was not surprising to also see a decline in occupancy. Changes in masked shrew occupancy rates in our plots may have been related to diet. This species has a very high metabolism and needs to feed almost constantly (Pearson 1947). McCay and Storm (1997) found that masked shrew abundance was positively related to invertebrate abundance in moist environments. A similar effect may have occurred in our study, as vegetation removal can reduce invertebrate biomass by up to 82% in hay fields (Zalik and Strong 2008). The absence of a similar response by the other small mammals in our study suggests that their energy requirements are not as extreme as that of masked shrews, and therefore they may be able to cope with either increased foraging rates or lesser invertebrate prey availability.

Of the 153 masked shrews captured in our study, 60 (39%) died in the traps, and this was likely related to their high metabolic needs (Little and Gurnell 1989). To determine whether trapping mortality had an effect on occupancy probabilities in our plots, we added a covariate for the effect of trap deaths (defined as the number of masked shrews found dead in each plot in each year) on local extinction and colonization probabilities in our best supported model. The resulting model was not competitive with

our best supported model (>2ΔAIC$_c$), which suggests that occupancy rates of masked shrews in our plots were unaffected by trap deaths.

As abundance data was only available for *Microtus*, deer mice, and northern short-tailed shrews, future studies would do well to collect enough data on the other 4 species/genera of small mammals in our study so that similar abundance analyses for these less common small mammals could be performed. This might be achieved by increasing the number of trapping occasions per year, the number of traps set per night, or the number of plots. An abundance analysis of masked shrews would be of particular value in determining the magnitude of probable decline in abundance of this species following increased harvest percentage. Additionally, determining population viability of *Microtus* following declines due to harvesting may be of value, as small mammal populations are able to rebound fairly rapidly following disturbances (Litt and Steidl 2011).

MANAGEMENT IMPLICATIONS

Native grasslands planted for biofuels should be harvested with an intensity that allows for a maximum amount of vegetation removal while at the same time causes the least amount of detriment to small mammal populations. Harvesting intensities of varying pattern (i.e. strip or block) that include up to 75% harvest are unlikely to cause significant declines in populations of *Microtus*; however, full harvest led to population reductions in our study. Conversely, deer mice are likely to increase in abundance with increased vegetation harvest, and thus the loss of one species may be partially accommodated by the gain of another. Other species of small mammals on our study plots seemed to be unaffected by biomass harvest, although masked shrews may become less prevalent with

increased harvest percentages. To prevent small mammals such as *Microtus* and masked shrews from becoming locally extirpated, we recommend that land managers work with biofuel harvesters to retain unharvested sanctuaries where these small mammals can persist despite biomass harvest.

Table 1. Best supported a priori models of species richness and abundance of *Microtus*, northern short-tailed shrews, and deer mice in experimental native grassland biofuel plots in western Minnesota, 2009-2012. Models are ranked according to ΔAIC_c.

	Microtus	Northern short-tailed shrews	Deer mice	Species richness
Year[a]*Harvest	0.000	-	1.633	-
Year+Harvest	-	-	1.420	-
Year*Pattern	-	-	-	-
Year+Pattern	-	-	-	-
Year*Harvest+Year*Pattern	-	-	-	-
Year+Harvest+Pattern	-	-	-	-
Year(BACI)[b]*Harvest	1.854	1.969	-	1.556
Year(BACI+Harvest	-	0.000	0.000	0.000
Year(BACI)*Pattern	-	1.725	-	-
Year(BACI)+Pattern	-	0.271	-	-
Year(BACI)*Harvest+ Year(BACI)*Pattern	-	-	-	-
Year(BACI)+Harvest+Pattern	-	1.991	-	1.475

[a] Year as a continuous covariate; 2009=0, 2010=1, etc.

[b] Year as a before-after control-impact covariate; pre-harvest=0, post-harvest=1

Table 2. Parameter estimates (β) and standard errors (SE) in best supported models of species richness and abundance of *Microtus*, northern short-tailed shrews, and deer mice in experimental native grassland biofuel plots in western Minnesota, 2009-2012.

	Microtus		Northern short-tailed shrews		Deer mice		Species richness	
Parameter	β	SE	β	SE	β	SE	β	SE
Intercept	2.882	0.253	-0.870	0.320	0.383	0.348	1.048	0.104
Year[a]	0.440	0.134						
YearBACI[b]			1.897	0.233	0.771	0.261	0.219	0.092
%Harvest	-0.496	0.328	-0.182	0.247	0.848	0.387	-0.180	0.108
Year:%Harvest	-0.074	0.188						
YearBACI:%Harvest								
PatternStrip[c]								
Year:PatternStrip								
RegionWC	-0.806	0.183	0.575	0.260			-	-
RegionSW	-1.045	0.186	0.908	0.237	-	-	-	-
Dispersion parameter	0.268	0.124	0.817	0.112	0.325	0.359	1.000	

[a] Year as a continuous covariate; 2009=0, 2010=1, etc.

[b] Year as a before-after control-impact covariate; pre-harvest=0, post-harvest=1

[c] Block harvest=0, strip harvest=1

Table 3. Models of the effects of time (t), species (sp), percent harvest (H), and harvest pattern (P) on occupancy (ψ), local extinction (ε), colonization (γ), and detection (p) probabilities of 7 species/genera of small mammals in experimental native grassland biofuel plots in western Minnesota, 2009-2012. Models are ranked according to differences in Akaike's Information Criterion corrected for small sample size (AIC_c).

Model	AIC_c	ΔAIC_c	w_i[a]	K[b]	Dev[c]
ψ(t+sp),ε(t+sp*H),γ(t+sp*H),p(t+sp)	4839.866	0.0000	0.99709	116	4589.068
ψ(t+sp),ε(t+sp*H*P),γ(t+sp*H*P),p(t+sp)	4852.139	12.2730	0.00216	126	4577.821
ψ(t+sp+),ε(t+sp),γ(t+sp),p(t+sp)	4854.246	14.3803	0.00075	104	4631.246
ψ(t+sp),ε(t+sp*P),γ(t+sp*P),p(t+sp)	4879.241	39.3755	0.00000	114	4633.108

[a] Relative likelihood of current model (*i*) based on AIC_c value.

[b] No. parameters.

[c] Model deviance.

Table 4. Maximum likelihood estimates (logit link) of the effect of percent harvest (H) on local extinction (ε) and colonization (γ) probabilities for the model ψ(t+sp),ε(t+sp*H),γ(t+sp*H),p(t+sp) for occupancy, local extinction, colonization, and detection probabilities of 7 species/genera of small mammals in experimental native grassland biofuel plots in western Minnesota, 2009-2012.

Parameter	Estimate	SE	95% CI
H Microtus ε	8.043	5.911	-3.542 to 19.63
H Blarina ε	-0.256	0.871	-1.964 to 1.451
H Zapus ε	-0.630	1.163	-2.909 to 1.649
H Mustela ε	1.941	1.009	-0.037 to 3.919
H Spermoph ε	0.156	0.815	-1.442 to 1.754
H Sorex ε	1.926	0.721	0.513 to 3.340
H Peromys ε	-0.341	1.101	-2.499 to 1.817
H Microtus γ	-52.78	247.4	-537.7 to 432.1
H Blarina γ	-0.750	0.830	-2.376 to 0.877
H Zapus γ	4.958	3.934	-2.753 to 12.66
H Mustela γ	1.018	1.255	-1.442 to 3.478
H Spermoph γ	0.517	1.185	-1.805 to 2.839
H Sorex γ	-3.074	1.129	-5.287 to -0.862
H Peromys γ	3.568	1.497	0.634 to 6.503

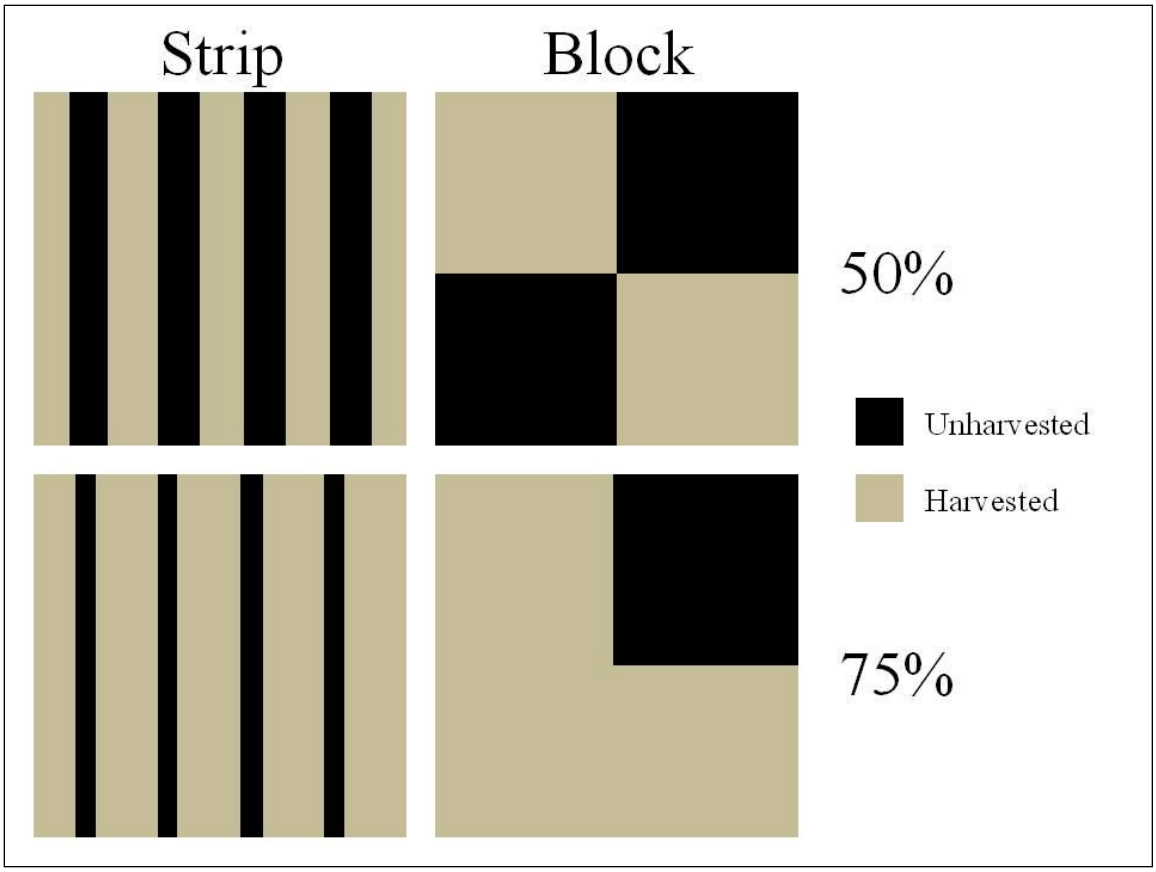

Figure 1. Differences between 50% and 75% harvest plot treatments in experimentally harvested native grassland biofuel plots in western Minnesota, USA, 2009-2012.

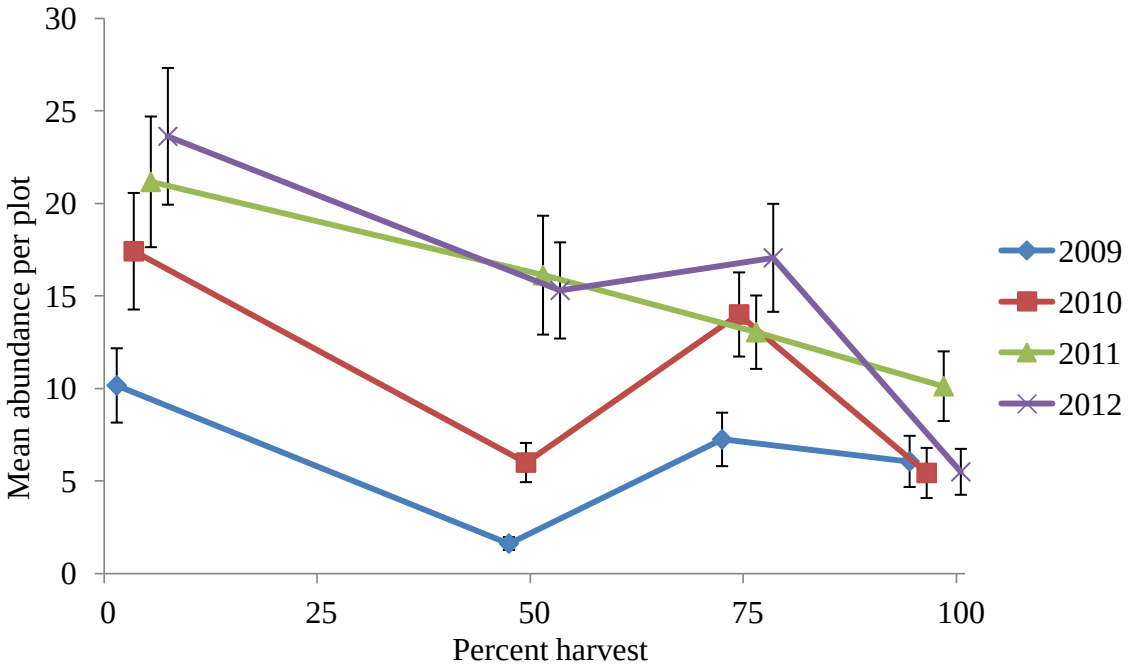

Figure 2. Mean abundances of *Microtus* per plot ± 95% confidence intervals in relation to percent harvest of native grassland biofuel plots in western Minnesota, USA, 2009-2012.

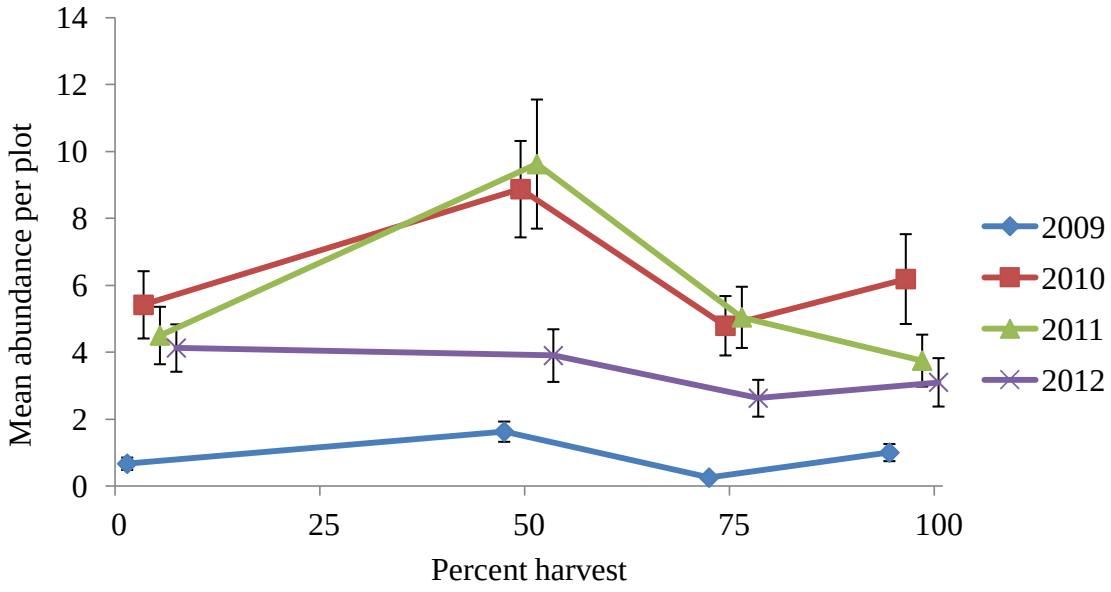

Figure 3. Mean abundances of northern short-tailed shrews per plot ± 95% confidence intervals in relation to percent harvest of native grassland biofuel plots in western Minnesota, USA, 2009-2012.

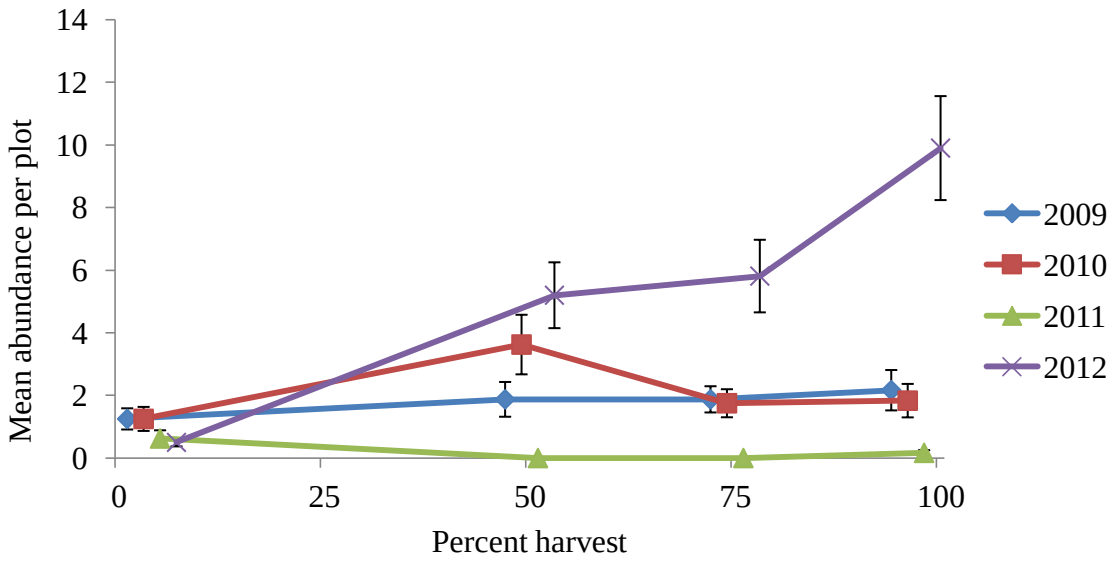

Figure 4. Mean abundances of deer mice per plot ± 95% confidence intervals in relation to percent harvest of native grassland biofuel plots in western Minnesota, USA, 2009-2012.

CHAPTER 3: IMPLICATIONS FOR LANDSCAPE-LEVEL MANAGEMENT

The results of our study suggest that harvesting native prairie vegetation causes little detriment to grassland birds and small mammals at the community level. Some species did show either declining or increasing abundance with increasing percent harvest, whereas other species were unaffected. Land managers would benefit bird and small mammal populations by identifying what species or genera were present prior to harvesting vegetation.

Once grasslands have been selected for biofuel production, land managers need to decide how much vegetation to harvest. Full harvests may potentially displace species that favored taller, dense vegetation in our study. Therefore a middle level of harvest may represent the most appropriate option. Somewhere in the 50% to 75% harvest range should satisfy this requirement, as it may leave enough residual standing vegetation to provide habitat for tallgrass songbirds and small mammals while at the same time provide shorter vegetation for the species in our study that showed a preference for harvested fields. Full harvests may be preferable if tallgrass species like sedge wren and common yellowthroat are absent to begin with, or if management plans include creating more habitat for species like grasshopper sparrow that favor shorter grass.

The context of grassland patches within the larger landscape should be considered, and a metapopulation analysis would be an important part of determining whether biofuel harvesting represents a viable conservation strategy in any given region. Complete harvest of a prairie may not actually detriment any grassland songbird or small mammal species, even a species like sedge wren that showed declines in our study, if that prairie was not originally a source of productivity (Pulliam 1988, Amarasekare and

Nisbet 2001). Yet if that prairie was a productive patch for sedge wrens, a decline in abundance resulting from vegetation removal might result in actual detriment to the species in that particular region. Grasslands should thus be evaluated on whether they exhibit source or sink characteristics for species of management concern prior to harvesting vegetation.

CRP fields have been identified as important sources of productivity for some grassland bird species (McCoy et al. 1999). Therefore, disturbance to these prairies may be harmful to species like Henslow's sparrow that have benefitted from the restored grassland vegetation (Roth et al. 2005). If a given prairie is a source of productivity, then, the greatest benefit to grassland birds may occur with the least amount of vegetation harvested. Conversely, if that prairie is a population sink hosting a non-breeding or dispersing group of songbirds, greater harvest percentages may cause little detriment to the overall population in the region if the birds are not actually breeding there.

Home range size is important to consider here as well. Birds displaced from vegetation removal in harvested fields may be able to relocate to non-harvested grasslands in the vicinity, resulting in no net loss in the regional population. Particularly if the distance between the disturbed and undisturbed grasslands is within the home range size of the given species (Wiens 1973, Knapton 1978, Burns 1982, Fletcher and Koford 2003) detriment may be unlikely as the birds are able to make frequent daily forages between patches.

The typical home ranges of voles, shrews, and mice are much smaller than those of birds (Harvey and Barbour 1965, Wolff 1985, George et al. 1986). For *Microtus*, which declined in abundance following harvesting of vegetation in our study, this

suggests that these small mammals may not be able to relocate to undisturbed grasslands as quickly following disturbance, potentially putting them at greater risk to predation (Birney et al. 1976, Tattersall et al. 2000). Particularly if the nearest undisturbed grasslands are beyond the home range size of *Microtus*, vegetation removal may be extremely detrimental if the harvested grassland was a source for vole productivity. Thus, when small mammals are part of a grassland management plan, leaving some portion of the prairie unharvested could potentially provide enough habitat for vole persistence.

Further studies would do well to collect data on demographic parameters of songbird and small mammal populations in native grassland vegetation experimentally harvested for biofuels. Demographic information might enable land managers to identify population sources and sinks (Balent and Norment 2003), thereby greatly reducing the potential for negative impacts on regional populations from harvesting vegetation. Furthermore, variables including nesting success, recruitment, and return rates might provide insight on the quality of biofuel grasslands following harvesting regardless of the observed effects on abundance and occupancy analyzed in our study (Gill et al. 2006, Zalik and Strong 2008).

Ideally, landowners would manage agriculturally unproductive marginal lands by planting native grassland vegetation, and the vegetation from these lands could be profitably harvested for biofuels (Fletcher et al. 2011). With current incentives in place, however, it is unlikely that grasslands will be planted solely for the purpose of biofuel harvest (Fargione et al. 2009). Rather, grassland biofuel production may more realistically offer supplemental benefits from CRP lands. Therefore policymakers that

promote this agricultural program can include biofuel production as a way for landowners to profit off land set aside for wildlife use.

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APPENDIX A.—Comparison of area-based surveys to 5-minute point counts in detecting grassland birds in experimental native grassland biofuel plots in western Minnesota, 2012. Mean ($\bar{x}$) and 95% confidence intervals (CI) are given, and all differences listed are significant ($P < 0.05$)

	Area-based surveys		5-minute point counts		Difference[a]	
	$\bar{x}$	95% CI	$\bar{x}$	95% CI	$\bar{x}$	95% CI
Total individuals (grassland species only)	12.24	10.72 to 13.77	6.78	5.96 to 7.61	38.1	29.8 to 46.3
Total grassland species	3.92	3.58 to 4.26	3.11	2.79 to 3.42	17.4	10.7 to 24.0
Total individuals (all species)	14.65	12.96 to 16.34	7.91	6.99 to 8.82	40.7	32.6 to 48.8
Total all species	4.81	4.36 to 5.26	3.68	3.32 to 4.03	19.2	12.9 to 25.5

[a] Values listed are percents

In area-based surveys, the observer began along the periphery of a plot 50m from a randomly predetermined corner and walked an "S" pattern through the plot, generally keeping at least 50m from the edge of the plot until completion of the third parallel transect upon reaching the opposite end (Fig. 1a). When more elongate plots were surveyed, the walking pattern changed to more of a "U" shape with only 2 parallel transects (Fig. 1b).

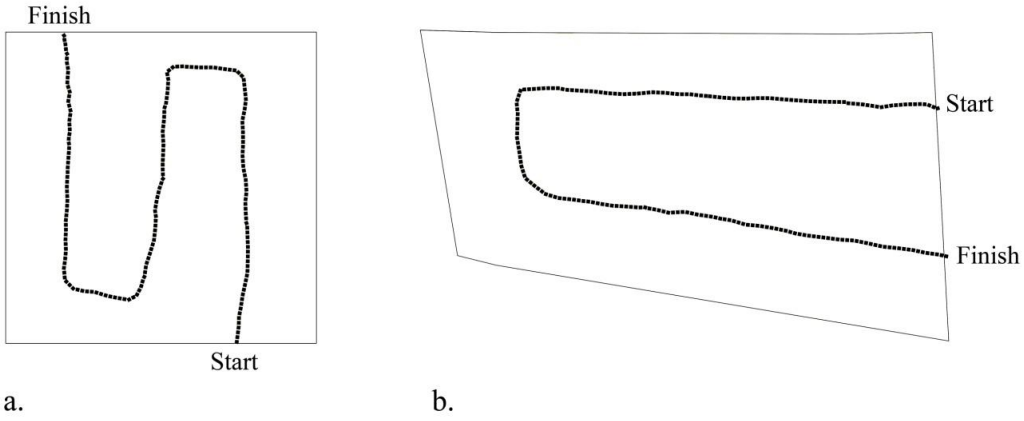

Fig. 1.—Variations in pattern walked in plots for area-based surveys. Pattern walked was dependent on plot shape

In 2012, 5-minute point counts were conducted in the center of each plot during the area-based surveys. Any bird occurring in the plot (excluding flyovers) was counted. Birds recorded during the area-based surveys were only included as detected during the point counts if they were observed during the 5-minute period. Conversely, birds detected during the point counts were not included in the area-based surveys unless they were also detected during the area-based surveys.